Journal of Hydrology X 2 (2019) 100006

Contents lists available at ScienceDirect

Journal of Hydrology X

journal homepage: www.elsevier.com/locate/hydrolx

Research papers

Restoration of blanket peat moorland delays stormflow from hillslopes and reduces peak discharge

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ARTICLE INFO

Article history:
Received 22 July 2017
Revised 29 October 2018
Accepted 22 November 2018
Available online 26 November 2018

Keywords:
Flood risk management
Natural flood management
Re-vegetation
Gully blocking
Water tables
Peatland hydrology

ABSTRACT

Over the past 15 years there has been a proliferation of projects aiming to restore the structure and function of UK upland blanket mires, primarily by revegetation of bare peat and the blocking of erosion gullies. These restoration measures have potential to alter stormflow responses and contribute to Natural Flood Management, but their impacts on storm hydrographs are poorly quantified. This paper reports a before-after-control-intervention (BACI) study from three experimental headwater micro-catchments in the South Pennines (UK) representing the first rigorous experimental assessment of the impact of blanket peat restoration on catchment runoff. We evaluate the hydrological impacts of two standard restoration interventions; revegetation of bare peat, and revegetation of bare peat with additional gully blocking. Following revegetation there was a significant decrease in depth to water table and an increase in the prevalence of hillslope overland flow production. There were no significant changes in storm runoff coefficient following either restoration treatment. Storm hydrographs following revegetation had significantly longer lag times (106% increase relative to the control), reduced peak flows (27% decrease relative to the control), and attenuated hydrograph shapes. With the addition of gully blocking the effect is almost doubled. Lag times increased by a further 94% and peak flows reduced by an additional 24% relative to the control. We argue that the primary process controlling the observed changes in storm hydrograph behaviour is retardation of overland stormflow due to increased surface roughness. The significant changes to lag times and peak flow provide evidence that the restoration of degraded headwater peatlands can contribute to Natural Flood Management and the reduction of downstream flood risk, subject to wider catchment scale effects and sub-catchment storm hydrograph synchronicity.

1. Introduction

Approximately 500,000 km^2 (12%) of global peatlands are severely degraded through human activity (Joosten, 2016). There has been growing interest in the hydrological processes controlling runoff from both degraded and restored peatlands (e.g. Price et al., 2016), including studies from cutover peats in Europe (Kløve and Bengtsson, 1999) and North America (e.g. Shantz and Price, 2006; Price and Ketcheson, 2009), and ditched or eroded blanket peats in North America (e.g. Price, 1992), the UK and Ireland (Burke, 1975; Holden and Burt, 2003; Holden et al., 2006; Luscombe et al., 2015). In the UK, headwater catchments are characterised by extensive blanket peat cover and have been subject to significant climatic and anthropogenic pressures (Bonn et al., 2009; Ramchunder et al., 2009; Clark et al., 2010) This has led to widespread ecosystem degradation in the form of erosion, drainage, pollution, and wildfire damage (Evans and Warburton, 2007; Parry et al., 2014). Upland blanket mires are therefore amongst the most damaged ecosystems in the UK with many peatland headwaters severely eroded. Large areas of bare peat and extensive erosional gully networks are common, including the North and South Pennines (Tallis, 1997; Garnett and Adamson, 1997), north and mid-Wales (Yeo, 1997; Ellis and Tallis, 2001), and Scotland (Grieve et al., 1994), with peatland erosion reported across 10–30% of the total UK blanket peat area (Evans and Warburton, 2007).

Blanket peatlands are hydrologically 'flashy' systems. In hydrologically intact systems, water tables are typically close to the

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https://doi.org/10.1016/j.hydroa.2018.100006
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ground surface (Evans et al., 1999), so that soil water storage is limited and rapid saturation excess overland flow is generated in response to significant rainfall events. Stream flow responds rapidly to rainfall events, producing relatively short hydrograph lag times and high peak flows relative to total storm runoff volumes (Evans et al., 1999; Holden and Burt, 2003).

Peatland degradation and erosion through loss of vegetation cover and/or gully development further increases the flashiness of stream flow response. Where vegetation is removed, surface roughness decreases, leading to increased hillslope overland flow velocities and faster delivery of hillslope drainage into channels (Holden et al., 2008). Bare peat surfaces may also develop hydrophobic properties (Eggelsmann et al., 1993; Evans et al., 1999) and be subject to surface compaction by raindrop action. This can reduce infiltration rates and increase infiltration excess overland flow production in high intensity rainfall events. The formation of gully networks increases drainage density, hillslope-channel connectivity and catchment drainage efficiency (Evans and Warburton, 2007). Peat erosion can therefore result in flashier storm hydrographs and higher storm-flow peaks, which have been linked to increased flood risk downstream (Baird et al., 1997; Grayson et al., 2010).

Over the past 15 years restoration of upland blanket peatlands in the UK has been extensive (Evans et al., 2005; Wallage et al., 2006; Armstrong et al., 2010; Parry et al., 2014), including landscape-scale restoration through the re-vegetation of bare peat and the blocking of erosion gullies (Anderson et al., 2009). Recent studies of the effects of restoration have focussed on carbon release (e.g. Dixon et al., 2013), vegetation recovery (e.g. Cole et al., 2014), and sediment dynamics (e.g. Shuttleworth et al., 2015), but relatively little is known about the effects of restoration on hydrological behaviour.

Natural Flood Management (NFM) describes the restoration of natural hydrological functions in damaged systems with the aim of reducing downstream flood risk (Dadson et al., 2017). Restoration of eroding peatlands has the potential to modify hydrological functioning, through changes in storm-flow runoff generation processes, runoff pathways and catchment storage (c.f. Acreman and Holden, 2013). Consequently, there is growing interest in the extent to which blanket peat restoration may regulate storm flows to downstream areas (e.g. Bain et al., 2011). Spatial averaging, which occurs in catchments, means that NFM benefits have been difficult to evidence in large catchments (>20 km^2) (Dadson et al., 2017). However, flood risk in upland catchments is commonly associated with the flashy response of small headwater systems (Wilkinson et al., 2013).

Plotscale experimental work by Holden et al. (2008) demonstrated the potential importance of vegetation related changes in surface roughness as a control on runoff velocities, and Grayson et al. (2010) reported correlations between long term vegetation change and changes in hydrograph form in naturally re-vegetating peatlands. Peatland re-vegetation may therefore be beneficial to NFM through changes in stormwater storage and/or the attenuation of flow. Gully blocking may also reduce runoff through the addition of pool storage or reduction in channel flow velocities due to increased channel roughness. These considerations suggest that peatland restoration can delay and/or reduce stormflow from headwater catchments. However, eroded blanket peats also have depressed water tables (Daniels et al., 2008; Allott et al., 2009), and there has been concern that raising water tables through restoration may reduce hillslope storage and increase runoff, as observed by Shantz and Price (2006) at a restored peat extraction site in Canada. Despite the large-scale implementation of peatland restoration, the impacts of re-vegetation and gully blocking on runoff have not been quantified (Parry et al., 2014). A more complete understanding of the impact of restoration on hillslope hydrology is required to determine the potential for peatland restoration to deliver NFM benefits.

This paper investigates changes in hydrological behaviour associated with blanket peat restoration by re-vegetation of bare peat and gully blocking, using three micro-catchments situated on the Kinder Plateau, Peak District National Park (PDNP), UK. This study has two objectives: i) to quantify the impacts of peatland re-vegetation on water tables and overland flow; ii) to quantify the impacts of peatland re-vegetation and gully blocking on storm hydrograph behaviour. These objectives provide the structural sub-headings used in the following Methods and Results sections. The Discussion section then reflects on the processes responsible for the observed changes, and how our findings contribute to wider debates surrounding the role of peatland restoration in NFM.

2. Study site and experimental design

2.1. Field area

The field experiment took place on the Kinder Scout Plateau in the Southern Pennines, UK. Kinder Scout represents one of the most severely eroded areas of blanket peat in the UK (Tallis, 1997), characterised by networks of erosion gullies and (prior to restoration) extensive areas of bare peat flats (Pilkington et al., 2015). Peat depths of 2–2.5 m overlie a sandstone bedrock from the Millstone Grit Series (MGS) (Wolverson Cope, 1976) and fine-grained head deposits of weathered MGS shales (Rothwell et al., 2005). Mean monthly temperatures recorded at the nearby Upper North Grain weather station vary between 13.2 °C (July) and 1.6 °C (February), mean annual rainfall is 1313 mm, and the prevailing wind direction is WSW (254°) (Clay and Evans, 2017).

In 2011/12, an 84 ha area on the north side of Kinder Plateau was selected for peatland restoration as part of the *Making Space for Water* demonstration project (Pilkington et al., 2015). Approximately 34% (28 ha) of this area consisted of severe gully erosion and bare peat. Extant vegetation cover prior to restoration was dominated by *Vacciunium myrtillus-Empetrum nigrum* heath on higher elevation peat hags, with some additional areas of *Eriophorum angustifolium*.

2.2. Restoration activities

The restoration treatments followed protocols developed, and applied in landscape scale restoration across the south Pennines by the Moors for the Future Partnership (Buckler et al., 2013). The restoration methodology and revegetation outcomes are summarised here. Full details of treatments for the study micro-catchments, including application rates, fertilizer doses and seed mix composition are provided in Pilkington et al. (2015).

Restoration by re-vegetation of bare peat uses lime, seed, and fertilizer and a cut heather mulch to establish a nurse crop, mainly composed of amenity grasses to provide initial ground cover. This stabilises the peat surface and provides the conditions for longer-term succession of native peatland plant species (see Anderson et al., 2009). Revegetation on Kinder consisted of five principal treatment stages (Pilkington et al., 2015):

(i) heather brash was spread by hand on areas of bare peat in March 2011;
(ii) granulated lime was applied on 20th July 2011 by helicopter (suspended hopper);
(iii) granulated NPK (nitrogen, phosphorus, and potassium) fertilizer was applied by helicopter on 21st July 2011;
(iv) a treatment of seeds of amenity grasses, local grasses and dwarf shrubs were applied by helicopter on 21st July 2011;

(v) Maintenance treatments of NPK fertilizer were made by helicopter in June 2012 and July 2013.

Restoration by gully blocking involves installing c. 0.5 m high stone dams composed of millstone grit cobbles (75–200 mm diameter), across the width of the gully in the main gully stem c. 6–7 m apart. Timber dams are also used in smaller tributary gullies constructed with a 38 mm deep 'V notch' cut into the top board to promote flow over the centre. On Kinder, stone dams were installed in winter 2011–12, and timber dams were installed in February-April 2012.

Post-restoration establishment of vegetation cover between 2011 and 2014 is reported in Pilkington et al. (2015). Aerial photography indicates that areas of bare peat cover across the site decreased by 76% between 2009 and 2014. At restored sites, quadrat surveys conducted in 2010 and 2014 showed change from zero vegetation cover to: 39% grass cover (including the amenity grasses *Lolium perenne*, *Festuca ovina*, and *Agrostis castellana* present in the applied seed mix), 27% Acrocarpus moss spp., 11% *Calluna vulgaris*, 6% Plearocarpous moss spp., 3% *Rumex acetosella*, 4% *Polytricum* spp and 2% Liverwort spp (Pilkington et al., 2015). There was also 34% cover of dead plant material. Almost complete cover by nurse crop had therefore established during this period, including on the floors of gullies, with remaining areas of bare peat confined largely to the steep gully walls. No significant changes in vegetation cover were observed at a 150 × 200 m area to the east of the Plateau that was left unrestored as a control (Pilkington et al., 2015).

2.3. Experimental design

We use a before-after-control-impact (BACI) design, using data generated over a five-year period from three micro-catchments. Three micro-catchments were selected to have comparable geometry and erosion and gully characteristics (Table 1), using DEM's (2 m resolution LiDAR), gully maps (Evans and Lindsay, 2010) and field verification (Fig. 1). One site (F) was left bare to act as a control, while the other two (O and N) revegetated during the restoration period. Gully blocks were also installed at site N; a total of 17 stone dams were installed along a 120 m section of the main channel and 20 wooden dams were installed in smaller tributary gullies.

Intensive monitoring started in June 2010 and covered a pre-restoration period of 15 months (2010–11), and 32 months post-intervention (2012–2014). Water tables and overland flow production were monitored to help understand changes to storage and runoff pathways, and rainfall and channel flow were monitored to investigate stormflow behaviour. Due to the costs of landscape- scale manipulations and restrictions in the availability of suitable field sites, it was not possible to replicate site conditions, leading to a pseudoreplicated design. This is a common challenge in landscape manipulation experiments and associated statistical analysis (Davies and Gray, 2015; Colegrave and Ruxton, 2018), and we have accounted for it by considering changes in behaviour relative to the control over multiple storm events.

Table 1
Micro-catchment site data.

Treatment type	Control	Re-vegetation	Re-vegetation and gully blocking
Micro-catchment ID	F	O	N
Location of catchment outlet (UK NGR)	408,972 389,442	408,262 389,464	408,234 389,464
Catchment area (m^2)	7008	4468	7096
% Gully area[a]	32.9	22.9	28.5
% Bare peat in non-gullied areas[b]	55	52	48
Max elevation (m)	618	617	619
Min elevation (m)	612	611	611
Mean catchment hill slope (degrees)	6.6	6.2	6.5

[a] Derived from 2 m^2 resolution LiDAR elevation data using the method of Evans and Lindsay (2010).
[b] Derived from 2009 air photography.

3. Methods

3.1. Field monitoring of water tables and overland flow

Depth to water table (DtWT) and the occurrence of overland flow were recorded to evaluate changes following re-vegetation. Measurements were taken manually at weekly intervals, between September and November 2010 (i.e. pre-restoration) and between and September and December 2014 (i.e. 3 years after restoration).

DtWT was determined using clusters of 15 dipwells located randomly within 30 × 30 m areas of the peatland (after Allott et al., 2009). Three dipwell clusters were established at the control (bare peat) site and three clusters within the two re-vegetated micro-catchments, giving a total of 45 dipwells for each of the control and revegetation treatments. All dipwells were located at least 2 m away from gully edges so that localised drawdown of water tables in proximity to erosion gullies (Daniels et al., 2008; Allott et al., 2009) was not a factor. Each dipwell comprised a 1 m length of polypropylene waste pipe (internal diameter 30 mm) with perforation holes drilled at 100 mm intervals. Dipwells were driven into pre -prepared boreholes. DtWT was measured relative to the ground surface.

The occurrence of overland flow was detected using crest-stage runoff traps (Holden and Burt, 2003) co-located with the dipwell clusters. Three clusters of traps, each containing nine tubes within a 1 m^2 plot, were located in the control area, and three clusters in the treatment catchments. Tubes were sunk into the peat surface with their entry holes flush with the peat surface. During sampling, the number of tubes containing water was recorded before 'wet' tubes were emptied to reset the cluster for the subsequent week of sampling. The overland flow quotient (OFQ) was calculated as the proportion of tubes recording overland flow within the sampling period.

3.2. Field monitoring of hydrograph behaviour

V-notch weirs and pressure transducers were installed at the catchment outlets. Pressure transducers (logging at 10 min intervals) recorded the depth of water (mm) flowing over the v-notch weir, which was subsequently converted to discharge and normalised to catchment area to facilitate comparison across catchments ($L\,s^{-1}\,km^{-2}$). Rain gauges at each site monitored rainfall at 10-min intervals. Rainfall and discharge data are available for each catchment from June 2010 to September 2011 (pre-restoration), and April 2012 to December 2014 (post-restoration). Operational issues led to periods where no data were collected for some sites, resulting in gaps in the record. This resulted from combinations of: (i) severe climatic conditions, including ice and severe low temperatures, leading to occasional equipment failure (ii) delays in servicing due to site access restrictions; and (iii) episodes of sedimentation impacting stilling pools.

For each catchment, the available rainfall and runoff data were collated. Hydrographs were extracted for all rainfall events where: (i) total rainfall exceeded 4 mm; and (ii) rainfall occurred as a discrete event with a single associated discernible main peak in discharge. Complex multi-peak hydrographs were excluded.

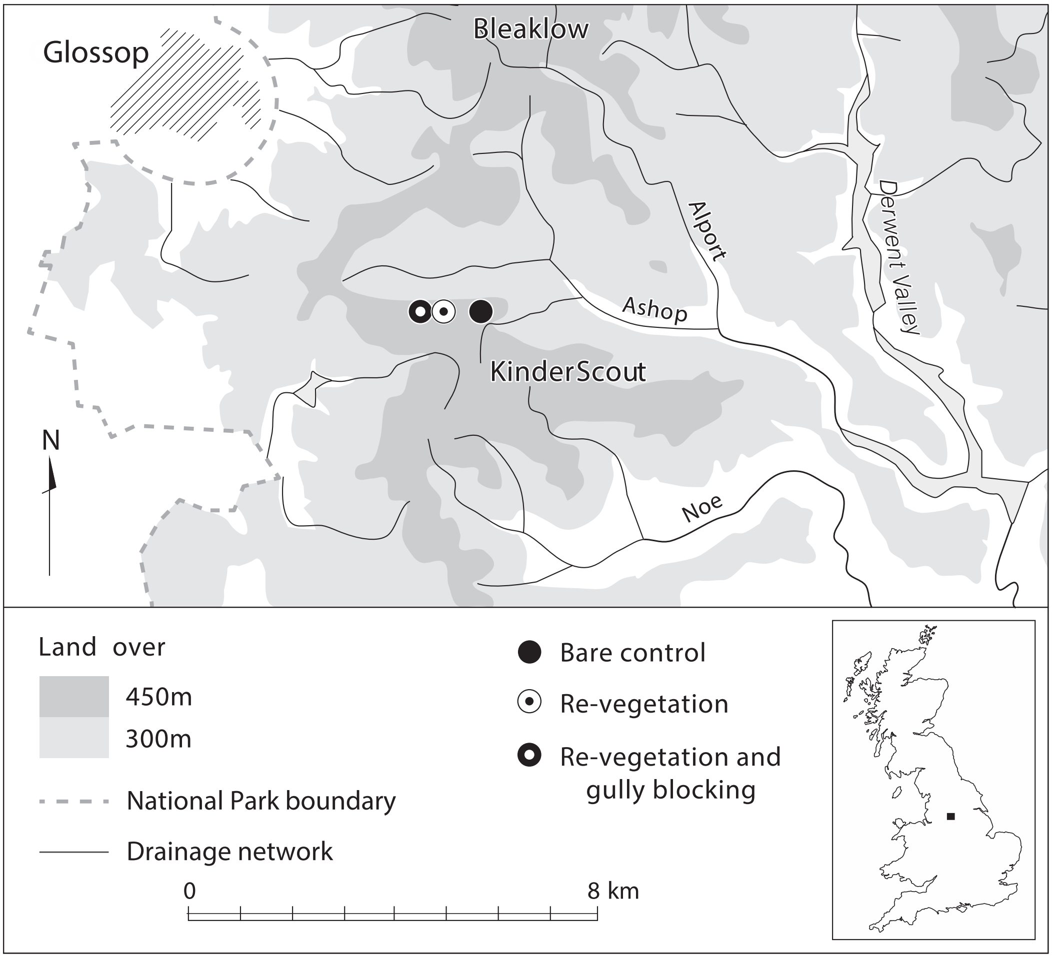

Fig. 1. Location of the study catchments.

The rainfall and runoff data from these hydrographs were used to calculate four key metrics: (i) lag time between peak rainfall and peak flow (*lag*); (ii) peak storm discharge (*peakQ*); (iii) Hydrograph Shape Index (*HSI*) (the ratio of peak storm discharge to total storm discharge, a measure of hydrograph intensity); and rainfall runoff coefficient (*C*). Storm-flow characteristics are influenced strongly by the intensity and duration of rainfall, so two rainfall characteristics were also derived to compliment the hydrograph metrics: (i) total rainfall (mm); and (ii) maximum rainfall intensity over a 10-min interval (mm h^{-1}).

3.3. Hydrograph data quality control

Data for a total of 506 hydrographs were extracted (152 storms for catchment F, 187 for O, and 167 for N). Runoff and rainfall metrics for these storms are summarised in Appendix A. The full dataset covers a total of 329 storm events. However, this includes 223 storms where hydrographs fitting the strict selection criteria could only be extracted for a single site. There were 68 storm events where hydrographs could be extracted for all three catchments.

As storm-flow characteristics are influenced by antecedent conditions and the nature of rainfall events (Evans et al., 1999), the mismatch in storm events in the complete data set could lead to substantial bias when comparing metrics between catchments. By analysing the 204 hydrographs derived from the 68 storms events for which metrics could be extracted for all three catchments, runoff behaviour resulting from similar rainfall and antecedent conditions could be compared directly. This reduced dataset allows for a strict and robust comparison of the data, and is the primary dataset used for all subsequent statistical analysis of hydrograph metrics.

There was still considerable 'noise' in the reduced dataset, due to the range of rainfall behaviours and antecedent conditions; total rainfall per event ranges from 4 to 56 mm, and maximum event rainfall intensity ranges from 1.8 to 54 mm h^{-1}, leading to a wide range of runoff responses in the storm-flow metrics. By standardising the metrics derived at the treatment catchments against the control catchment we can differentiate responses due to restoration treatment from natural variation. This was achieved by deriving the *relative difference* (treatment minus control) between the metrics produced by control and treatment sites.

3.3.1. High magnitude events

To assess the potential utility of peatland restoration as an NFM measure in upland catchments, it is important to assess the degree to which changes in runoff delivery are maintained in high magnitude events. In particular, if hillslope and channel storage control runoff, then NFM efficacy may be reduced in large storms since storage as a proportion of storm runoff would be minimised. Data from the ten biggest pre-restoration and ten biggest post-restoration storms (total event precipitation) were compared. Analysis of the large storm subset was based on standardised metrics from the two treatment sites relative to the control.

3.4. Statistical analyses

Many of the variables of interest do not follow a normal distribution (overland flow, rainfall, hydrograph metrics), so non-parametric tests of difference were employed to determine the statistical significance of the influence of restoration. Where data were available for each year of the study, Kruskal Wallis 1-way ANOVA were used to investigate year-on-year changes following restoration. Pairwise comparisons were applied post-hoc using adjusted Mann-Whitney U tests, to assess where any significant differences lie. Where only one year of post-restoration data was available (water table, overland flow), or the data set had been reduced to 'before' and 'after' data, Mann-Whitney U tests were used to investigate the effects of restoration. All relationships were tested at the 95% level ($p \leq 0.05$).

In using non-parametric analyses, we were unable to assess the additional benefit of gully blocking statistically, as there is no non-parametric equivalent of a 2-way ANOVA which would allow us to examine the effect of two factors ("before/after restoration" and "treatment type") in an unbalanced dataset. Any impacts of gully blocking are discussed in terms of additional magnitude of change relative to re-vegetation alone.

4. Results

Figures show the *relative difference* between the treatment and control sites (treatment minus control), before and following restoration. Positive values on the y-axis therefore indicate that the metric of interest is greater at the treatment site than at the bare control, while negative values indicate the opposite. All parameters are discussed in terms of their median value.

4.1. Impacts of peatland re-vegetation on water tables and overland flow

In 2010, prior to restoration, water tables were closer to the surface at the treatment site than at the control site (DtWT were 307 and 345 mm respectively; Table 2). The relative difference in DtWT ($DtWT_{rel}$) was 27 mm (Fig. 2a). In 2014, DtWT at the control site was 342 mm, comparable to the 2010 value, while DtWT at the treatment site was 293 mm, shallower than DtWT observed pre-restoration. $DtWT_{rel}$ had increased to 59 mm. This represents a significant relative decrease in DtWT of 35 mm (i.e. 9% relative to the control) at the treatment site following re-vegetation ($p = 0.010$, Mann-Whitney U).

In 2010, both the treatment and control sites produced comparable amounts of overland flow (median OFQ of 0.19 and 0.22 respectively; Table 2). The relative difference in OFQ (OFQ_{rel}) was highly variable around zero (Fig. 2b). Median OFQ_{rel} was negative (−0.07), demonstrating that prior to restoration, the treatment site was less productive of overland flow than the control. In 2014, the relationship was reversed with a positive median OFQ_{rel} value of 0.11. Although this increase in overland flow at the treatment site is not statistically significant ($p = 0.065$, Mann-Whitney U), there has been a clear shift in behaviour. After treatment, OFQ_{rel} on all bar one measurement day was positive, indicating that the re-vegetated site was producing consistently more overland flow than the control.

4.2. Impacts of peatland re-vegetation and gully blocking on hydrograph behaviour

4.2.1. Annual data

Descriptive statistics for the four key hydrograph metrics at the three micro-catchments are summarised in Table 3, and the relative differences between the treatment and control sites (treatment minus control) are presented in Fig. 3. These relative differences are referred to as lag_{rel}, $peakQ_{rel}$, HSI_{rel}, and C_{rel}. Groupings of statistically similar years (based on Kruskal-Wallis 1-way ANOVA) are represented by lower case letters. The data suggest that restoration has had an immediate effect on three out of the four metrics at both treatment sites. Lag_{rel} increased and both $peakQ_{rel}$ and HSI_{rel} were reduced immediately after restoration. There was no consistent change in C_{rel}, with post-restoration values similar to pre-restoration values in two of the three post-treatment years. Following the pronounced step change in lag_{rel}, $peakQ_{rel}$, and HSI_{rel} in 2012, there are no subsequent directional trends apparent in any of the metrics.

Lag_{rel} shows the clearest evidence of a consistent step change in behaviour following restoration. At both treatment sites, lag_{rel} in 2010/11 (i.e. before restoration) fall into Group a, while all subsequent years fall into Group b (Fig. 3a and b). Lag_{rel} pre-restoration was therefore significantly different to lag_{rel} post-restoration, and lag_{rel} was statistically similar in the three years following restoration. Similar groupings can be seen for HSI_{rel} (Fig. 3e and f). A step change is apparent but less pronounced for $peakQ_{rel}$, with the two treatment sites producing different groupings. At the re-vegetated site (O), the two years following treatment are distinct from the pre-restoration period (Group b), but 2014 produced similar $peakQ_{rel}$ to the pre-restoration period (Group a) (Fig. 3c). At the re-vegetated and blocked site (N), the three years post-treatment are similar (Group b) but the year immediately after restoration (2012) can also be grouped with the pre-restoration period (Group a) (Fig. 3d). The high post-restoration $peakQ_{rel}$ in 2014 coincides with anomalously high relative C_{rel} values at the same site (Fig. 3g), indicating that variation in $peakQ_{rel}$ is non-random.

The simplest explanation of the trends observed in the data is a step change in lag_{rel}, $peakQ_{rel}$ and HSI_{rel}, in response to restoration.

Table 2
Summary statistics for depth to water table (DtWT) and overland flow quotient (OFQ) based on weekly data gathered between September and December in the years 2010 (pre-treatment) and 2014 (3 years post-treatment).

		Depth to Water Table (mm)		Overland Flow Quotient	
		Control	Treatment	Control	Treatment
	N	11	11	11	11
2010	Median	345	307	0.22	0.19
	Maximum	422	439	0.78	0.93
	Q3	364	323	0.61	0.28
	Q1	255	257	0.11	0.07
	Minimum	198	204	0.07	0.00
2014	Median	342	293	0.04	0.15
	Maximum	484	428	0.81	0.52
	Q3	391	325	0.09	0.33
	Q1	307	256	0.02	0.09
	Minimum	286	242	0.00	0.04

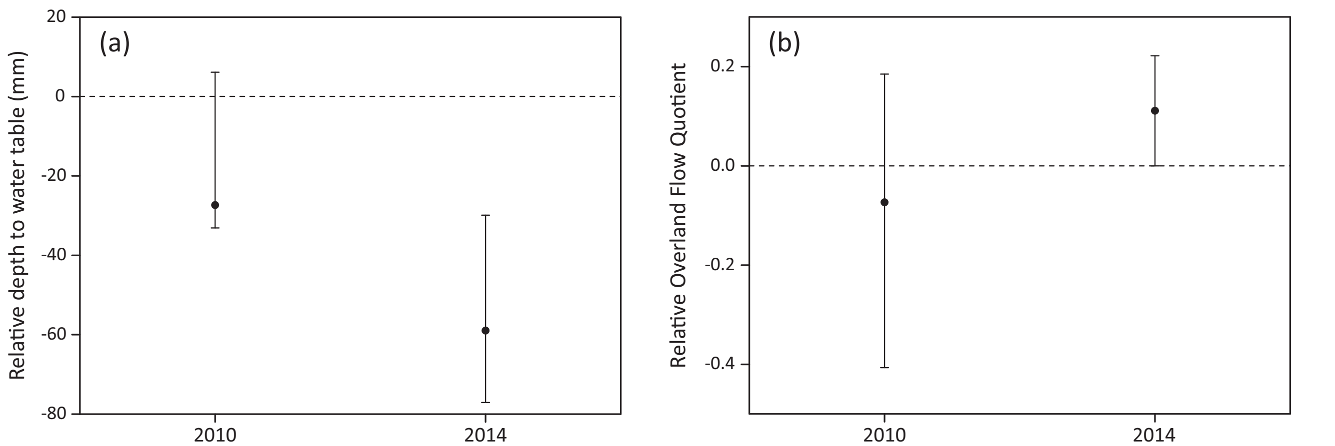

Fig. 2. Median relative difference in (a) depth to water table (DtWT), and (b) overland flow quotient (OFQ) based on weekly data gathered between September and December in the years 2010 (pre-treatment) and 2014 (3 years post-treatment). Positive values indicate that the metric is greater at the treatment site than at the bare control, while negative values indicate the opposite. Error bars represent the interquartile range.

Table 3
Annual summary statistics for the four key hydrograph metrics for 'paired' storms.

		2010–11			2012			2013			2014		
		F	O	N	F	O	N	F	O	N	F	O	N
	N	20	20	20	16	16	16	19	19	19	13	13	13
Lag (min)	Median	30	15	20	20	42.5	60	15	25	35	25	35	45
	Maximum	90	50	60	75	115	155	75	85	355	195	275	235
	Q3	30	35	35	45	73	100	15	35	95	35	45	125
	Q1	20	10	20	15	20	38	5	25	20	15	25	35
	Minimum	10	0	10	0	10	20	5	5	5	5	15	15
Peak Storm Discharge ($L\,s^{-1}\,km^{2}$)	Median	490	750	610	510	420	560	450	520	220	880	940	450
	Maximum	4970	4010	2510	3170	3080	2930	6340	6230	3180	1690	1720	1440
	Q3	1270	1390	1470	1030	960	730	1090	940	470	1270	1250	610
	Q1	260	360	330	280	220	210	310	270	60	330	360	190
	Minimum	50	80	50	110	120	10	120	100	0	160	290	20
HSI	Median	0.16	0.20	0.16	0.21	0.17	0.11	0.22	0.18	0.14	0.22	0.17	0.16
	Maximum	0.28	0.59	0.33	0.57	0.77	0.26	0.89	0.44	1.21	0.64	0.37	0.34
	Q3	0.20	0.23	0.19	0.35	0.26	0.16	0.33	0.28	0.30	0.41	0.26	0.24
	Q1	0.13	0.12	0.12	0.15	0.11	0.10	0.15	0.14	0.11	0.17	0.14	0.12
	Minimum	0.05	0.06	0.08	0.09	0.06	0.08	0.06	0.05	0.07	0.10	0.08	0.08
Rainfall Runoff Coefficient (%)	Median	40.3	48.5	40.9	31.4	31.1	34.2	28.7	32.0	19.4	24.4	44.1	28.8
	Maximum	71.0	79.6	66.3	52.3	58.5	57.1	67.9	60.5	62.2	59.1	85.4	60.0
	Q3	60.5	63.7	55.3	38.6	38.1	49.2	38.5	44.0	26.9	37.0	60.3	33.3
	Q1	20.3	30.5	21.3	20.1	21.1	24.1	17.2	18.6	6.9	17.6	33.2	20.5
	Minimum	5.8	8.7	3.7	6.6	8.2	0.4	7.7	6.5	0.2	6.8	11.7	0.3

Subsequent variability is interpreted as inter-annual noise resulting from variation across the storms available for analysis. Based on this interpretation, we combine the three years of post-restoration data into a single 'after' restoration dataset, to determine the average magnitude of the changes in hydrograph behaviour following restoration.

4.2.2. BACI analysis

Prior to treatment, hydrographs at the three sites behaved in a similar manner (Table 4): lag ranged between 15 and 30 min, PeakQ was between 490 and 750 $L\,s^{-1}\,km^{-2}$, HSI ranged between 0.16 and 0.20, and *C* was between 40 and 48%. There were no significant differences in hydrograph metrics at the three sites before treatment (Kruskal Wallis 1-way ANOVA, $p > 0.05$ for all parameters). However, it should be noted that Sites F and O were consistently at opposite ends of these ranges, with site O displaying flashier flow characteristics (shorter lag times, higher PeakQ), consistent with its smaller catchment area (Table 1).

Before restoration, lag_{rel} at site O (re-vegetation only) was −10 min (Fig. 4a); lag was 61% of that at the control site. Following restoration, lag_{rel} increased to 10 min and lag was on average 167% of the control. This represents a statistically significant increase in lag_{rel} of 20 min, or a 106% increase in lag relative to the control ($p < 0.001$, Mann-Whitney U). At site N (re-vegetation with additional gully blocks), prior to restoration lag_{rel} was 0 (i.e. lag was on average the same length as at the control site). Following restoration, lag_{rel} increased to 30 min, and lag was on average 300% of the control. This represents a statistically significant increase in lag_{rel} of 30 min, or a 200% increase in lag relative to the control ($p < 0.001$, Mann-Whitney U).

Prior to restoration, $peakQ_{rel}$ at site O was 131 $L\,s^{-1}\,km^{-2}$ (Fig. 4b); peakQ was 129% that of the control site. Following restoration, $peakQ_{rel}$ decreased to −12 $L\,s^{-1}\,km^{-2}$, indicating that peak discharges were roughly the same as the control. This represents a statistically significant decrease in $peakQ_{rel}$ of 143 $L\,s^{-1}\,km^{-2}$, or a 27% decrease in peakQ relative to the control ($p = 0.001$, Mann-Whitney U). At site N, in the pre-restoration

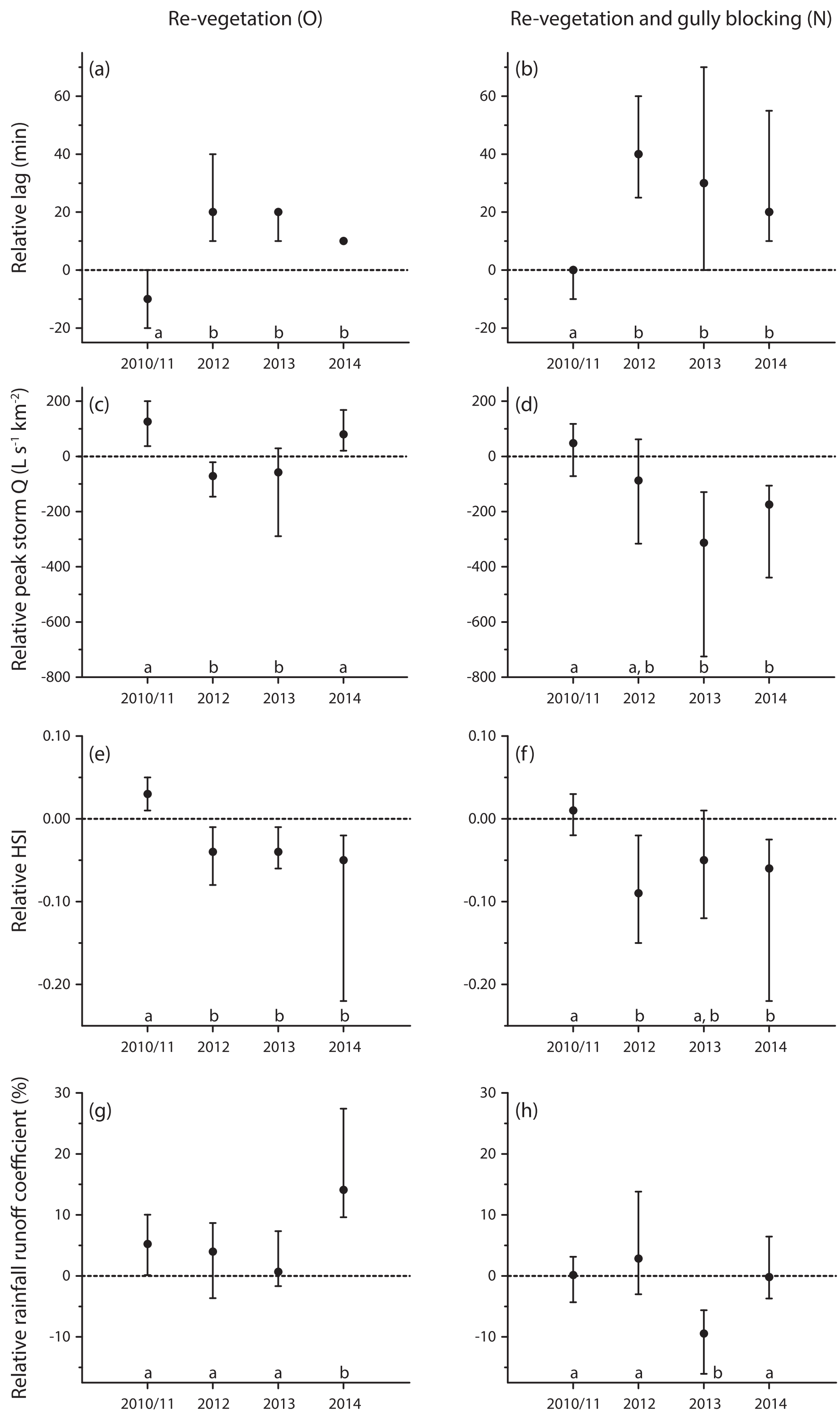

Fig. 3. Annual median relative differences between the treatment and control sites for key hydrography metrics: lag time (a and b), peak discharge (c and d), Hydrograph Shape Index (e and f), and percent runoff (g and h). Positive values indicate that the metric is greater at the treatment site than at the bare control, while negative values indicate the opposite. Error bars represent the interquartile range. Groupings of statistically significant years (based on Kruskal-Wallis 1-way ANOVA) are represented by lower case letters.

period peakQ_{rel} was 55 L s^{-1} km^{-2}; PeakQ was similar to the control site (108%). Following restoration, PeakQ_{rel} decreased to -200 L s^{-1} km^{-2}, and peakQ was only 57% of the control. This represents a statistically significant 255 L s^{-1} km^{-2} decrease in PeakQ_{rel}, or a 51% decrease in peakQ relative to the control ($p = 0.001$, Mann-Whitney U).

Table 4
Summary statistics for the four key hydrograph metrics for 'paired' storms before and after intervention.

		Before			After		
		F	O	N	F	O	N
	N	20	20	20	48	48	48
Lag (min)	Median	30	15	20	15	25	55
	Maximum	90	50	60	195	275	235
	Q3	30	35	35	35	50	110
	Q1	20	10	20	10	25	35
	Minimum	10	0	10	0	5	5
Peak Storm Discharge ($L\ s^{-1}\ km^2$)	Median	490	750	610	580	540	370
	Maximum	4970	4010	2510	6340	6230	3180
	Q3	1270	1390	1470	1090	1020	640
	Q1	260	360	330	290	310	140
	Minimum	50	80	50	110	100	0
HSI	Median	0.16	0.20	0.16	0.22	0.18	0.14
	Maximum	0.28	0.59	0.33	0.89	0.77	1.21
	Q3	0.20	0.23	0.19	0.38	0.28	0.22
	Q1	0.13	0.12	0.12	0.15	0.12	0.10
	Minimum	0.05	0.06	0.08	0.06	0.05	0.07
Rainfall Runoff Coefficient (%)	Median	40.3	48.5	40.9	29.5	34.3	25.6
	Maximum	71.0	79.6	66.3	67.9	85.4	62.2
	Q3	60.5	63.7	55.3	38.5	45.9	39.6
	Q1	20.3	30.5	21.3	18.1	23.9	17.7
	Minimum	5.8	8.7	3.7	6.6	6.5	0.2

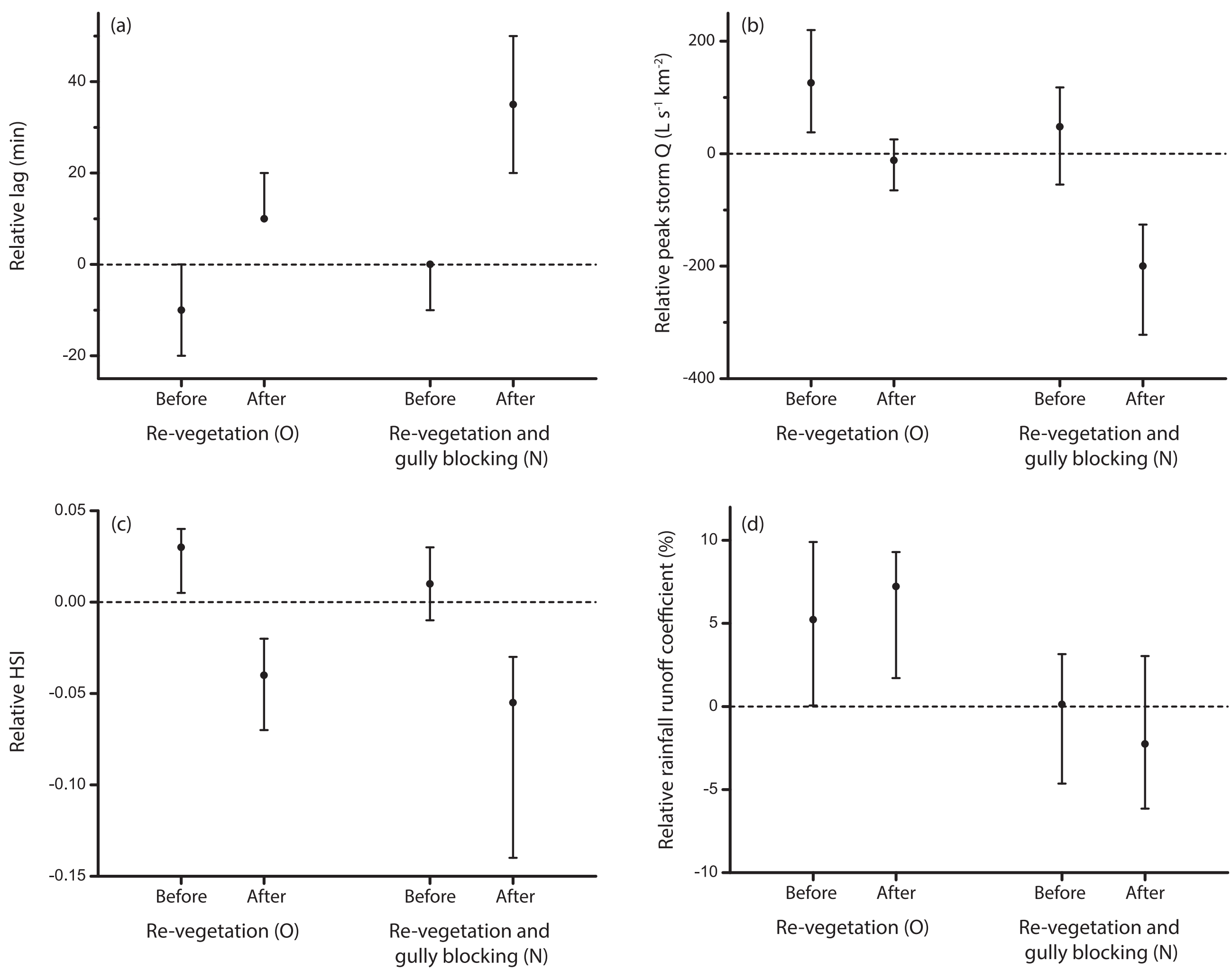

Fig. 4. Median relative differences between the treatment and control sites before and after restoration: (a) lag time, (b) peak discharge, (c) Hydrograph Shape Index, and (d) percent runoff. Positive values indicate that the metric is greater at the treatment site than at the bare control, while negative values indicate the opposite. Error bars represent the interquartile range.

During the pre-restoration period, HSI_{rel} at site O was 0.024 (Fig. 4c); HSI was 117% of the control site. Following restoration, HSI_{rel} decreased to −0.038, so HSI was 80% of the control. This represents a statistically significant decrease in HSI_{rel} of 0.062, or a 37% decrease in HSI relative to the control ($p < 0.001$, Mann-Whitney U). Following restoration at site N, HSI_{rel} was 0.005; i.e. HSI was similar to the control site (107%). Following restoration, HSI_{rel} decreased to −0.059, and HSI was 69% of the control. This represents a statistically significant decrease in HSI_{rel} of 0.064, or a 38% decrease in HSI relative to the control ($p < 0.001$, Mann-Whitney U).

There was no change in C_{rel} at either site following restoration ($p_N = 0.676$ and $p_O = 0.888$, Mann-Whitney U). Before intervention, the C_{rel} at site O was 4.5% indicating that it was more productive of runoff than the control; following restoration C_{rel} increased slightly to 7.2%. C_{rel} at site N was 0.36% prior to restoration, indicating that runoff production was similar to the control site; following restoration, C_{rel} fell slightly to −2.3%. This represents shifts in C_{rel} of 2.7% and −2.6% at sites O and N respectively. However, it is clear from the graph in Fig. 4d that post-restoration C_{rel} at both sites is well within the range of pre-restoration values, so it is unsurprising that this variation is not statistically significant.

Installing gully blocks in addition to re-vegetation as part of the restoration treatment increased lag_{rel} by a further 10 min (i.e. lag increased by an extra 94% relative to the control), and decreased $peakQ_{rel}$ by an additional 112 L s^{-1} km^{-2} (i.e. peakQ decreased by a further 24% relative to the control). However, the gully blocks did not have any additional effect on the magnitude of change in HSI_{rel}, which decreased by 37 and 38% at the treatment sites following restoration.

4.2.3. High magnitude storms

The magnitude of the effects of restoration practices were also investigated for the largest storms in the dataset (summarised in Table 5) to test if the effects of the intervention were still evident under more extreme rainfall conditions. Storm magnitudes ranged between 11 and 36 mm total precipitation before restoration, and 15 and 56 mm after intervention (Appendix B). The relative differences between the treatment and control sites (treatment minus control) for each metric are shown in Fig. 5. All parameters discussed in this section are median values.

As in the main dataset, there was a statistically significant increase in lag_{rel} at both of the treatment sites during high magnitude storms (Fig. 5a; $p_O = 0.019$ and $p_N = 0.035$, Mann-Whitney U). While the scale of change at site O was similar to that of the full dataset, the magnitude of change at site N was considerably less than when considering all storms. For large storms, lag_{rel} at site O increased by 25 min (109% increase in lag), similar to the main dataset where lag_{rel} increased by 20 min (106% increase in lag). However, at site N lag_{rel} only increased by 10 min during large storms, representing a 25% increase in lag, much less than the 200% increase in lag in the full dataset. There was also a statistically significant decrease in $peakQ_{rel}$ at both treatment sites during large storms (Fig. 5b $p_O = 0.019$ and $p_N = 0.035$, Mann-Whitney U). The magnitude of change in peakQ at site O was smaller than that of the full dataset (17% versus 27%), while at site N it was similar to the main dataset (56% versus 57%). HSI_{rel} was also reduced for large storms (Fig. 5c) by 28% at site O and 26% at site N. This was less than when considering all storms (37% and 38%), and this shift in hydrograph shape post restoration was not statistically significant ($p_O = 0.075$ and $p_N = 0.143$, Mann-Whitney U). Unlike the full data set, C_{rel} was reduced at both treatment sites during large storms following restoration (Fig. 5d), but these reductions were not statistically significant ($p_O = 0.971$ and $p_N = 0.123$, Mann-Whitney U).

5. Discussion

5.1. The impact of restoration on runoff generation

Restoration has had a pronounced effect on the hydrology of the peatland headwater catchments, producing marked changes in water table depth, runoff production, and storm-flow behaviour. Restoration by re-vegetation raised water tables by 35 mm after three years, 're-wetting' the treated areas, which in turn increased the incidence of overland flow relative to un-treated sites. Re-vegetation has also had an immediate and significant impact on storm hydrograph characteristics, increasing lag times by 106%, and decreasing peak storm discharge by 27% and HSI by 37%. Gully blocking enhances the benefits of re-vegetation, with lag times

Table 5
Summary statistics for the four key hydrograph metrics for the 10 highest magnitude storms before and after intervention.

		Before			After		
		F	O	N	F	O	N
N		10	10	10	10	10	10
Lag (min)	Median	25	10	20	15	25	30
	Max	90	50	60	195	275	235
	Q3	30	20	27.5	30	42.5	35
	Q1	20	10	13	15	25	18
	Min	10	0	10	5	5	5
Peak Storm Discharge (L s^{-1} km^{2})	Median	1130	1240	1260	1350	1120	570
	Max	4970	4010	2510	6340	6230	3180
	Q3	1650	1860	1490	2010	1610	1350
	Q1	530	740	760	1000	640	350
	Min	220	490	410	150	360	20
HSI	Median	0.14	0.16	0.12	0.19	0.17	0.17
	Max	0.28	0.31	0.33	0.89	0.44	0.56
	Q3	0.19	0.19	0.17	0.30	0.31	0.30
	Q1	0.11	0.11	0.11	0.12	0.11	0.10
	Min	0.07	0.09	0.08	0.06	0.05	0.07
Rainfall Runoff Coefficient (%)	Median	52.2	57.5	50.6	43.6	44.3	29.9
	Max	71.0	69.7	66.3	67.9	81.8	62.2
	Q3	61.1	63.7	63.4	52.1	56.8	50.4
	Q1	39.1	41.2	45.5	14.3	28.3	12.8
	Min	12.6	34.8	27.0	6.8	6.5	0.3

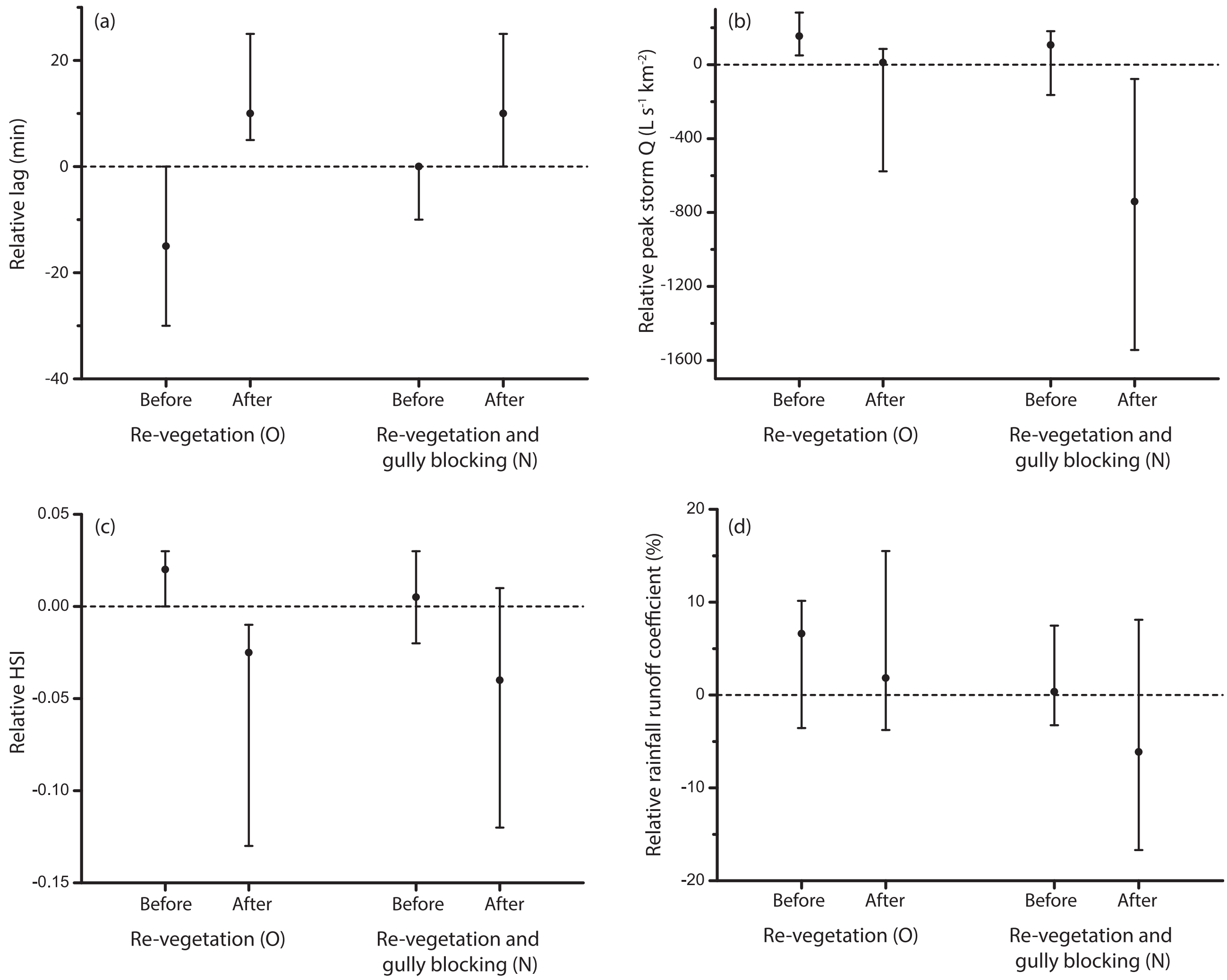

Fig. 5. Median relative differences between the treatment and control sites for the largest 10 storms before and after restoration: (a) lag time, (b) peak discharge, (c) Hydrograph Shape Index, and (d) percent runoff. Positive values indicate that the metric is greater at the treatment site than at the bare control, while negative values indicate the opposite. Error bars represent the interquartile range.

increased by a further 94%, and peak storm discharge reduced by an additional 24% relative to the control. However, gully blocking does not appear to alter the 'flashiness' of stormflow, as HSI was reduced by a similar proportion at both of the treatment sites. Neither of the treatments have had any impact on the proportion of storm event rainfall that becomes storm discharge (C). The changes to hydrograph behaviour post-restoration are still evident during large storms (Fig. 5), albeit to a lesser extent for some parameters, indicating that the changes in runoff delivery are maintained in high magnitude events. The observed hydrological impacts of restoration in peatland headwaters therefore have the potential to alter downstream stormflow behaviour and reduce flood risk.

We acknowledge that our results are based on a limited number of sites, and that replicating our experiments at different locations would strengthen our findings. However, our data corresponds with findings from observational and modelling studies, providing confidence that our results are more generally valid. For example, Grayson et al. (2010) observed c.20% reductions in peak discharge due to natural revegetation in a blanket peatland in the North Pennines. Gao et al. (2016) modelled the impacts of vegetation cover on riparian strips on peatland stormflow in blanket peatlands in England and Wales, using comparable rainfall intensities to those observed in our study, and found that bare riparian zones increased peak flow by up to 20%, while *Sphagnum*-covered ground reduced peak flow by up to 13%. Similarly, Pan and Shangguan (2006) observed 14–25% reductions in runoff by adding grass to bare soil in a plot study of loessial loam in Yangling, China.

5.2. Process controls – What might be causing these changes in runoff generation?

5.2.1. Water tables and overland flow

Re-vegetation has raised water table depth and increased incidence of overland flow over a relatively short period (c.3 years), but not yet to levels comparable with intact peatlands (c.f. Evans et al., 1999; Holden et al., 2006). Water table recovery has been widely documented in peatlands where artificial drains have been blocked (e.g. Shantz and Price, 2006; Wilson et al., 2010; Haapalehto et al., 2011; Holden et al., 2011; Menberu et al., 2016). However, the present study provides the first example of re-vegetation alone improving water table condition in blanket peatlands.

Water table rises under re-vegetation could be driven by a range of potential mechanisms. The insulating properties of newly established vegetation cover may reduce evaporative losses (cf. Grayson et al., 2010). Price et al. (1998) found that net radiation and soil heat flux were greater over bare peat when compared to mulched surfaces, attributing this to the mulch's higher albedo, so the change from dark bare peat to higher albedo re-vegetated surfaces is likely having a similar effect. Alternatively, root penetration may increase infiltration or vegetation may increase microtopographic storage. There may also have been structural changes in the peat matrix over time, reducing hydrophobicity and increasing the peat's ability to retain water. Investigations of the peat matrix combined with analyses of net radiation and evapotranspiration data are needed to distinguish between these hypotheses.

Incidences of overland flow increased following re-vegetation. Holden and Burt (2003) show that saturation-excess overland flow is the dominated runoff mechanism in an intact peatland in the Northern Pennines, and work by Evans et al. (1999) suggests that this is linked to the reactive nature of peatland water tables to precipitation. Increased overland flow is therefore consistent with the declines in depth to water table discussed above. However, re-vegetation has not restored runoff conditions to those of the intact site reported in Holden and Burt (2003), suggesting that incomplete water table recovery (constrained by the fact that topography is not returned to pre-erosion form by the restoration; Holden et al., 2006) limits runoff recovery.

Despite the observed increase in overland flow and apparent reduced storage capacity due to rising water tables, percentage runoff values have not changed following restoration, indicating that there is no significant change in catchment storage during storm events. Peatlands characteristically have low specific yields (Price, 1996), so that the change in storage associated with small changes in water table in this instance appears to be within measurement noise.

5.2.2. Hydrograph response

The runoff coefficient shows no change following re-vegetation implying that hillslope storage is not altered. Despite there being no change in long term storage, the rate of delivery of runoff has been reduced, as illustrated by increased lag times and attenuated hydrograph shapes. Grayson et al. (2010) observed similar changes in hydrograph behaviour at a naturally revegetated peatland site in the North Pennines. This is likely due to increased surface roughness provided by the newly established vegetation (cf. Holden et al., 2008; Pan and Shangguan, 2006).

Gully blocking enhances the impacts of re-vegetation on peak discharge and lag time, but there was no significant change in runoff coefficient in the blocked catchment, indicating that there has been no gain in storage through ponding behind gully blocks (cf. Evans et al., 2005). This is perhaps surprising but indicates that the additional changes in stormflow hydrographs associated with the gully blocking in this study are driven by the introduction of large scale roughness elements to the channel. It should be noted that this does not mean that gully blocking lacks the potential to increase catchment storage. The findings presented here are based on a single catchment with a particular arrangement of blocks, and the optimal approaches to block design and spacing are yet to be determined. Adjusting the number of blocks, and their spacing and design has the potential to further attenuate stormflow and increase catchment storage (Milledge et al., 2015). The trajectory of hydrograph response in gullied systems will also require evaluation, given the relatively short time period represented in the current analysis and the potential for longer-term effects associated with gully infilling and revegetation. Comparison with hydrological data emerging from other gully blocked peat systems, such as the high-elevation peats of the Tibetan Plateau (Zhang et al., 2012, 2016), will be instructive.

5.3. Wider implications

The data we present, provide the first controlled catchment scale experimental evidence that hydraulic roughness controlled by vegetation cover drives the rainfall-runoff response in blanket peatlands (cf Grayson et al., 2010). The establishment of the nurse crop has had an immediate (i.e. within one growing season) impact on stormflow characteristics with no further trends in the subsequent years (Fig. 3). However, based on the work of Holden et al. (2008), it is reasonable to assume that surface roughness and hence the rainfall-runoff response will be further modified over longer time scales as the vegetation matures and natural blanket bog species return. The fact that the roughness effect dominates runoff (as opposed to storage) means that it persists in high magnitude storm events (Fig. 5). This is an important finding in terms of NFM, especially in headwater catchments where overland flow and flow depths are relatively shallow.

5.3.1. Restoration and land management

These findings have implications beyond simply re-vegetating areas of bare peat. *Sphagnum* is regarded as a 'keystone' species in peatlands (Rochefort, 2000; Gorham and Rochefort, 2003) due to its role in bog building and maintaining high water tables and acidic conditions, and its reintroduction is becoming a priority in blanket peat restoration initiatives. In plot experiments Holden et al. (2008) demonstrated that *Sphagnum* had the greatest impact on slowing overland flow velocities (only c.10% that of bare ground), so widespread reintroduction has the potential to make a major contribution to NFM, especially if strategically targeted in riparian zones (Gao et al., 2016). Similarly, other land management practices which alter vegetation cover, such as the creation of clough woodland, grazing (Anderson and Radford, 1994), and prescribed burning (Clay et al., 2009; Holden et al., 2015) may also impact downstream flood risk through surface roughness effects. Further work is needed to better quantify the effects of different land-covers and -uses at the catchment scale.

This study highlights the importance of identifying suitable control sites to underpin the results of short term (less than decadal duration) catchment studies. The use of appropriate control has removed substantial amounts of 'noise' in the data resulting from inter-annual variation in synoptic hydrometeorology. Without the control site, we may have incorrectly deduced a storage effect, as the raw data showed a reduction in *C* at the treatment sites post-restoration (Table 4); however, this is not the case when *C* is considered relative to the control (Fig. 4). Similarly, we observed lower incidence of overland flow in the treatment catchments post-restoration (Table 2), but a substantial increase in overland flow relative to the control catchment (Fig. 2). By assessing deviations from the control, we have been able to detect the magnitude of the effect of restoration, independent of synoptic conditions. The control component of the BACI design is therefore critical when assessing the impact of ecosystem restoration, to avoid misleading results and to understand the processes driving post-intervention change. This is especially important when considering the policy relevance of environmental science research (Bilotta et al., 2014, 2015), and should be considered an essential element of restoration monitoring.

5.3.2. Downstream flood risk

The significant post-restoration changes in hydrology observed in this study will reduce flood risk at the headwater scale. These headwater effects will propagate downstream, with the potential to reduce flood risk substantially at the wider catchment scale. Reduction in downstream flood risk will depend on two important scale factors. Firstly, the scale of restoration relative to the size of the catchment (Milledge et al., 2015), and the position of restoration works in the landscape (Gao et al., 2016). Secondly, the nature of catchment and sub-catchment geography and associated hydrograph synchronisation effects contributes to the wider catchment hydrograph (Pattison et al., 2014; Metcalfe et al., 2018). This is an important consideration. For example, Nutt and Perfect (2011) present evidence that a moorland improvement scheme designed to delay runoff in the Allen Water (Scotland) may have partially synchronised sub-catchment peak flows and so increased downstream flood risk. Conversely, similar planned moorland improvement to a different sub-catchment were shown to help desynchronise the runoff from this sub-catchment and decrease flood risk (Nutt and Perfect, 2011). We note that restored blanket peats are located typically at the extreme upper end of drainage and catchment networks, so in this context an increase in storm-flow travel times from these systems would generally be expected to reduce peak flows downstream.

The use of monitoring approaches to evaluate these scale effects, and to quantify the benefits of restoration on downstream flood risk reduction, is problematic. This is due to multiple influences on flow regimes in wider catchments and confounding factors, making it difficult to isolate the effects of restoration within empirical storm-flow datasets (Dadson et al., 2017). It is also extremely difficult to identify suitable control systems at the large catchment scale. However, the benefits of restoration effects on flood risk reduction at larger catchment scales can be quantified using hydrological models (e.g. Lane and Milledge, 2012). The results of the current study provide the basis for realistic and robust hydrological modelling of downstream flood risk change. The study has quantified changes in lag times and peak flows from headwaters associated with restoration, and has demonstrated the hydrological processes that underlie these effects. These two factors permit appropriate model formulation and calibration (e.g. Milledge et al., 2015), but further catchment scale studies are required to better inform modelling assessments.

6. Conclusion

This study has adopted a paired catchment approach within a BACI design to demonstrate that re-vegetation of bare peat and gully blocking reduces rates of hillslope runoff from blanket peatland. Water tables have become shallower and the incidence of overland flow has increased, but this has not significantly affected the volume of storm runoff produced. Peak discharges are reduced and lag times are increased, despite there being no overall change in catchment storage. This is consistent with reduced hillslope runoff velocities due to increases in surface roughness provided by the newly established vegetation and gully blocks.

Modifications in peatland vegetation cover and drainage, whether deliberate or as a consequence of changing land use and management regimes, will have consequences in terms of downstream flood risk. The significant magnitude of the changes detailed in this study demonstrates that there is a clear and convincing evidence base to develop the role of peatland restoration techniques within the NFM framework. Operationalising these findings will require upscaling of the evidence from this study, but the empirical data presented here, and the finding that runoff delivery rather than storage is key, provides a basis for modelling the potential impacts at larger scales.

Until now, investment in peatland restoration has been justified by reference to enhanced biodiversity and to the role of peatlands in carbon storage. Our findings suggest that these large-scale modifications of upland landscapes may also play a role in protecting communities from flooding, adding to the multiple beneficial ecosystem services these peatlands provide.

Conflict of interest

There are no conflicts of interest.

Acknowledgements

This work represents a contribution from the 'Making Space for Water' project funded by DEFRA's Multi-Objective Flood Management Demonstration Scheme, with additional funding from the Environment Agency. We are extremely grateful to Fiona Draisey and the Peak District National Park Rangers for the extensive field data collection, without which the analysis of water tables and overland flow would not have been possible. Thanks also go to: John Moore for field kit design and manufacture; Karen Eynon and Clare Brown for support in setting up the project; John Gorham, Jane Mellor, Rebecca Richards and Andrew Stimson for field support and data collection; and Clive Agnew, Gareth Clay, James Rothwell, and Claire Goulsbra for helpful discussions. We also thank the two anonymous reviewers and the journal editorial team for their constructive comments which improved the quality of the paper.

Appendix A

		2010–11			2012			2013			2014		
		F	O	N	F	O	N	F	O	N	F	O	N
	N	34	45	44	36	42	46	45	53	42	37	47	35
Lag (min)	Median	30	20	20	15	45	75	15	35	80	25	45	75
	Maximum	120	120	90	75	205	330	215	315	355	195	275	235
	Q3	32.5	30	40	25	76.25	133.8	30	75	157.5	45	75	125
	Q1	20	10	20	15	25	39	5	25	15	15	25	35
	Minimum	10	0	10	0	10	5	5	5	−5	5	5	15
Peak Storm Discharge ($L\ s^{-1}\ km^2$)	Median	400	710	420	430	310	300	410	280	210	410	360	350
	Maximum	4970	4010	2510	3170	3080	2930	6340	6230	3180	3160	1720	1440
	Q3	1190	1440	1290	1000	500	570	810	530	370	910	710	540
	Q1	200	260	220	220	190	180	200	120	20	220	240	120
	Minimum	50	30	50	50	70	10	60	40	0	60	10	10
HSI	Median	0.14	0.59	0.17	0.27	0.14	0.11	0.20	0.16	0.13	0.23	0.15	0.12
	Maximum	0.36	0.68	0.59	0.98	0.77	0.36	0.89	0.49	1.71	0.83	0.55	0.71
	Q3	0.20	0.26	0.27	0.46	0.21	0.14	0.32	0.24	0.32	0.41	0.23	0.17

Appendix A (*continued*)

		2010–11			2012			2013			2014		
		F	O	N	F	O	N	F	O	N	F	O	N
	Q1	0.11	0.21	0.12	0.18	0.10	0.09	0.13	0.10	0.10	0.17	0.09	0.08
	Minimum	0.05	0.12	0.07	0.09	0.06	0.04	0.06	0.05	0.06	0.10	0.04	0.05
Rainfall Runoff Coefficient (%)	Median	32.1	38.9	32.0	22.5	29.0	31.6	26.8	28.8	20.2	22.9	35.5	29.8
	Maximum	71.0	86.5	66.3	52.3	58.8	72.2	67.9	63.3	124.3	59.1	106.9	61.9
	Q3	50.1	57.5	51.9	31.6	39.2	45.7	40.6	41.4	33.6	33.6	48.0	46.4
	Q1	16.6	20.1	18.2	13.4	19.2	22.6	13.3	16.8	0.6	14.9	22.7	15.8
	Minimum	5.5	4.5	3.2	4.2	8.2	0.4	3.3	2.8	0.0	3.4	0.5	0.1

Appendix B

		Before	After
N		10	10
Total Precipitation (mm)	Median	13.3	19.7
	Maximum	35.9	55.7
	Q3	20.9	23.3
	Q1	12.7	16.4
	Minimum	11.0	15.3
Maximum Precipitation Intensity (mm h^{-1})	Median	10.02	15.3
	Maximum	18.96	53.88
	Q3	14.1	18.96
	Q1	6.84	10.38
	Minimum	6.12	4.92

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